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DESIGNING IMPACT ASSESSMENTS FOR EVALUATING ECOLOGICAL EFFECTS OF AGRICULTURAL CONSERVATION PRACTICES ON STREAMS[1]

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ABSTRACT: Conservation practices are regularly implemented within agricultural watersheds throughout the United States without evaluating their ecological impacts. Impact assessments documenting how habitat and aquatic biota within streams respond to these practices are needed for evaluating the effects of conservation practices. Numerous sampling protocols have been developed for monitoring streams. However, protocols designed for monitoring studies are not appropriate for impact assessments. We developed guiding principles for designing impact assessments of ecological responses to conservation practices. The guiding principles are as follows: (1) develop the hypothesis first, (2) use replicated experimental designs having controls and treatments, (3) assess the habitat and biological characteristics with quantitative and repeatable sampling methods, (4) use multiple sampling techniques for collecting aquatic organisms, and (5) standardize sampling efforts for aquatic organisms. The guiding principles were applied in designing a study intended to evaluate the influence of herbaceous riparian buffers on channelized headwater streams in central Ohio. Our example highlights that the application of our recommendations will result in impact assessments that are hypothesis-driven and incorporate quantitative methods for the measurement of abiotic and biotic attributes.

(KEY TERMS: agricultural streams; conservation practices; environmental impacts; experimental design; environmental sampling; aquatic ecology; Conservation Effects Assessment Project.)

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INTRODUCTION

Conservation practices were traditionally defined as methods for managing soil and water resources to improve agricultural production, but the current definition includes methods to reduce the environmental impacts of agriculture on terrestrial and aquatic ecosystems (Soil and Water Conservation Society, 2001). Specifically, conservation practices are land, water, and agronomic management practices designed to reduce erosion rates, improve water quality, and restore terrestrial and aquatic habitats in agricultural streams. Conservation tillage, riparian buffers, and wetland creation are just a few examples of practices that have been promoted and funded through various federal and state programs. Conservation practices have been regularly implemented within

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agricultural streams throughout the United States (U.S.) since the 1970s without pre- or postevaluations of the resulting impacts (Bernhardt *et al.*, 2005; Alexander and Allan, 2006). Evaluations of the physical and biological responses of agricultural streams to conservation practices are needed to document the impacts of these practices and their effectiveness in providing environmental benefits (Moore and Palmer, 2005; Vondracek *et al.*, 2005). Documentation of the ecological effects of conservation practices will provide guidance for developing management plans intended to reduce the impacts of agriculture on streams.

Introductory compilations of specific techniques for assessing habitat and aquatic biota within streams are available (Merritt and Cummins, 1996; Murphy and Willis, 1996; Bain and Stevenson, 1999; Hauer and Lamberti, 2006). Recommendations on technique selection and descriptions of their use are provided within the numerous sampling protocols that have been developed for rapid bioassessment and other types of monitoring (see Johnson *et al.*, 2001; NRCS, 2001; USEPA, 2002; Somerville and Pruitt, 2004; Stolnack *et al.*, 2005 for descriptions of >400 sampling protocols). Habitat assessment sampling protocols focus on the measurement of hydrology, geomorphology, or riparian habitat variables, while biological assessment sampling protocols concentrate on methods for sampling aquatic macroinvertebrates or fishes (Table 1). Various rapid bioassessment protocols are widely used by regulatory agencies, consulting firms, and private organizations within the U.S. (Rabeni *et al.*, 1999). These protocols are designed to produce a spatially extensive dataset capable of detecting impairments in streams using cost and time-saving techniques (Rabeni *et al.*, 1999), but this dataset will not be effective in identifying environmental factors responsible for the impairment (Maddock, 1999; Winger *et al.*, 2005). Thus, the use of these protocols within studies intended to evaluate the effects of specific management practices results in sacrificing inferential power to gain rapid applicability (Downes *et al.*, 2002). Additionally, these sampling protocols may not be suitable for evaluating the effects of specific conservation practices because of differences in the objectives between monitoring studies and impact assessments (Downes *et al.*, 2002).

TABLE 1. Percentage of Different Types of Sampling Protocols Summarized Within Selected Reviews of Sampling Protocols for Monitoring Biological and Habitat Characteristics of Streams.

Type of Sampling Protocol	Somerville and Pruitt (2004)	NRCS (2001)	Johnson *et al.* (2001)	USEPA (2002)
Biological				
Bacteria	0	0	2	0
Algae	2	3	7	31
Aquatic or terrestrial macrophyte	0	3	5	15
Phytoplankton or zooplankton	0	0	0	15
Aquatic macroinvertebrate	12	22	29	86
Fish	6	19	31	63
Amphibian	4	0	2	5
Waterfowl	0	0	0	3
Wildlife and other vertebrates	0	8	5	5
Habitat				
Chemical	20	11	51	2
Hydrology and instream habitat features	55	28	57	32
Geomorphology	73	44	42	23
Riparian	55	61	36	23
Watershed	20	17	14	0
Total reports summarized	51	36	96	65

Note: Column totals may exceed 100% because they were calculated based on the number of reports listed within a review and reports often contained descriptions of multiple sampling protocols.

Monitoring studies are often conducted to evaluate the status or condition of streams and rivers (Downes *et al.*, 2002). Monitoring studies are designed without *a priori* knowledge of the stressor or the source of impact and are intended to address the question: "*What is the status of the target populations/communities and habitat conditions, and how does the status change through time?*" Conversely, impact assessments are conducted to evaluate the influence of specific disturbances or management practices (Downes *et al.*, 2002) and are designed with *a priori* knowledge of the expected source of impact (i.e., the disturbance or conservation practice). Impact assessments are designed to answer the question: "*How does implementation of conservation practices influence the habitat and target populations/communities?*" An impact assessment would also attempt to determine the environmental factors and causal relationships responsible for the observed impact of conservation practices on the aquatic biota. It is possible that monitoring studies may fortuitously obtain data that enables the evaluation of the impacts of a conservation practice. However, in this case the evaluation of the effects of a conservation practice with a monitoring study is a matter of chance, whereas the evaluation with an impact assessment is a directed and proactive effort specifically designed to document the impacts.

The spatial and temporal scale of monitoring studies can differ from impact assessments. Monitoring studies are analogous to weather stations and often involve repeated measurements through time over large geographic areas to provide a time series representing the status of the target/populations and habitat conditions. Conversely, impact assessments are conducted at a variety of spatial and temporal scales depending on the research hypothesis and funding base (Engel and Voshell, 2002). Impact assessments with large spatial and temporal scales are considered the most robust and necessary for evaluating the impacts of conservation and restoration practices (National Research Council, 1992; Kondolf, 1995), but smaller scale studies can provide valuable information as long as their limitations are recognized.

Differences between monitoring studies and impact assessments highlight the need for guidance in designing impact assessments. Our objective is to provide five guiding principles that will assist others in designing impact assessments of ecological responses of agricultural streams to conservation practices. We first provide background information and describe the challenges faced in developing the guiding principles. Second, we explain each principle and defend their importance. We conclude with an example of how the guiding principles were applied in designing an evaluation of the influence of herbaceous riparian buffers on the habitat and aquatic communities within channelized headwater streams (i.e., drainage ditches).

FIGURE 1. Locations of Agricultural Research Service Conservation Effects Assessment Project Watershed Assessment Study Watersheds Within the United States. Watersheds symbolized by a black circle within a white circle are study sites where ecological assessments are being conducted.

TOWARDS THE DEVELOPMENT OF GUIDING PRINCIPLES

The guiding principles below resulted from an attempt to develop standardized sampling protocols for the Agricultural Research Service (ARS) component of the Conservation Effects Assessment Project (CEAP). The ARS component of CEAP consists of the Watershed Assessment Study that is intended to provide detailed findings on the influence of conservation practices from 14 watersheds in the U.S. (Mausbach and Dedrick, 2004) (Figure 1). The ARS research effort focuses mostly on evaluating water chemistry and hydrological responses to conservation practices. Ecological responses are currently being evaluated in only three watersheds (Figure 1) despite the critical need for this information due to the elevated threat level currently faced by stream ecosystems (Richter *et al.*, 1997; Karr *et al.*, 2000). Additionally, the national assessment of conservation practices as part of CEAP plans to use watershed models (Mausbach and Dedrick, 2004) to evaluate water chemistry and selected hydrological responses to conservation practices (Shields *et al.*, 2006a). These models are not capable of evaluating physical habitat responses (i.e., base-flow discharge, substrate types, large woody debris, water temperature) or biological responses (i.e., fish and macroinvertebrate community responses) (Shields *et al.*, 2006a). Ecological assessments (i.e., joint assessments of habitat and biological responses) have the potential to provide information for novel developments in these watershed models that would allow the national assessment of conservation practices to conduct a cross-disciplinary assessment of conservation practices through the evaluation of water chemistry, hydrology, physical habitat, and ecological responses. We initiated the development of a standardized protocol to facilitate the: (1) initiation of ecological research within ARS watersheds, (2) collaborative ecological research among watersheds, and (3) cross-disciplinary assessments of conservation practices.

Developing a standardized sampling protocol immediately proved to be a challenge. Twelve of the ARS watersheds are lotic ecosystems, while two of the watersheds are lentic ecosystems. Watershed sizes range from 850 to 827,000 ha. The watersheds are located in 11 states within the Northeast, Midwest, Southeast, and Northwest U.S. and encompass 11 ecoregions ranging from the Middle Atlantic Coastal Plain to the Northern Basin and Range Ecogregions (Omernik, 1987, 1995). The amount of cropland within each watershed ranges from 2 to 100% and land use in seven watersheds is predominately cropland (≥60%). Six of the ARS studies are evaluating the influence of nutrient management and/or riparian buffers. However, research within 13 ARS watersheds also includes assessments of one to three other conservation practices from among 12 types ranging from manure management to drainage management. We concluded that developing a standardized protocol that designated the use of specific techniques was not appropriate given the diversity of potential environmental conditions and the variety of conservation practices being evaluated. Instead, we focused our efforts on developing a set of guiding principles suitable for designing ecological assessments and selecting response variables and methods.

FIVE GUIDING PRINCIPLES

We developed five guiding principles that provide a framework for developing ecological assessments. The principles were developed based on our past experiences conducting ecological research in agricultural watersheds and guidance provided by Green (1979), Cooper and Barmuta (1993), and Downes *et al.* (2002). Our intention was to highlight aspects of experimental design that when used together would result in robust ecological assessments. Although laboratory and mesocosm experiments are feasible, we focus on issues related to designing impact assessments. One example of an impact assessment to evaluate conservation practices would involve an investigator sampling habitat and biota from streams with and without conservation practices before and after the implementation of the conservation practices. An investigator's level of control of experimental treatments within impact assessments may range from field experiments that involve pre-planned implementation of conservation practices within selected streams to those where the investigator lacks manipulative control and exercises control by selecting and comparing sites that differ in the presence and absence of a conservation practice.

Impact assessments are known by other names such as impact studies (Green, 1979), natural experiments (Diamond, 1983), comparative mensurative experiments (Hulbert, 1984), impact assessments (Stewart-Oaten *et al.*, 1986), intervention analyses (Eberhardt and Thomas, 1991), and impact monitoring (Downes *et al.*, 2002). Impact assessments are extremely valuable because it is difficult to reproduce the essential character of aquatic and terrestrial ecosystems in the laboratory, in mesocosms, or through numerical simulations. We feel impact assessments will yield the most realistic information regarding the ecological effects of conservation practices because these studies are conducted at spatio-temporal scales similar to that of the experimental treatment (i.e., conservation practices). We did not attempt to provide guidance relating to technical options such as selection of taxonomic group(s) to evaluate, which biotic sampling methods to use in specific ecoregions, and data analyses. Specific decisions regarding these issues are dependent on the hypothesis, experimental design, site specific factors, and/or available funds.

Develop the Hypothesis First

Our first recommendation is to develop a hypothesis first, and then select the response variables and sampling methods based on the hypothesis. We feel this is the most important recommendation because many past field studies evaluating the influence of anthropogenic habitat alterations did not develop hypotheses (Cooper and Barmuta, 1993; Bennett and Adams, 2004). Hypothesis development is a fundamental component of the scientific method that is applicable to all forms of environmental research. The hypothesis needs to be specific enough that it can provide a basis for selection of response variables and sampling methods (Kondolf, 1995). Those hypotheses that provide an explanation for the expected effect (observed pattern) will be the most useful and will result in studies that are capable of providing an explicit explanation for the impact (Jansson *et al.*, 2005). However, it is not always possible to identify causal mechanisms that regulate the influence of a conservation practice on the aquatic biota before beginning an evaluation. In these cases simple hypotheses can be developed based on the expectations as to which abiotic and biotic factors will be altered by the implementation of a conservation practice. The use of *a priori* hypotheses to select response variables and methods requires investigators to consider what variables need to be evaluated before beginning the field research. This should lead to cost-effective studies as it will prevent the measurement of response variables that are not relevant to the hypothesis.

Use Replicated Experimental Designs With Appropriate Controls and Treatments

We view the implementation of a conservation practice within a stream or its watershed as an experiment in which the conservation practice serves as the experimental treatment having the potential to alter dependent variables (i.e., habitat or biota) (Underwood, 1990). Additionally, in the context of impact assessments, streams without the conservation practice of interest serve as the control sites, and streams that contain or will contain the conservation practice serve as the treatment sites. Unfortunately, many assessments of agricultural conservation practices have failed to include a control stream into the experimental design (Galeone, 1999). Control streams are necessary to isolate the effect of conservation practices from other factors (Downes *et al.*, 2002). Control streams are similar to laboratory controls in that both serve as the untreated treatment for the experiment (Hulbert, 1984). Yet, control streams differ from laboratory controls in that environmental conditions of control streams cannot be regulated to reduce the influence of confounding factors, but environmental conditions can be regulated in laboratory controls (Hulbert, 1984). Hence, selection of control streams plays a critical role and they need to be selected to ensure the effects of conservation practices are not confounded by the inherent differences among control and treatment streams.

We also feel the *a priori* development of specific habitat criteria to guide the selection of control and treatment streams is a good practice. However, specific selection criteria will differ among experimental designs. Site selection within before–after-control-impact, before–after, and impact *vs*. control sites designs (Table 2) involves consideration of selected physical features to ensure the similarity among control and treatment streams. Conversely, site selection within space-for-time (Table 2) and gradient designs (Table 2) involves consideration of selected physical features that need to differ among streams to create the treatment effect.

An individual stream (i.e., tributary or subwatershed) is typically accepted as a replicate in stream ecology due to interdependency in abiotic and biotic conditions among sites within streams (Downes *et al.*, 2002). For example, considering three sites within one control stream and three sites within one treatment stream (i.e., the one having the conservation practice implemented) as a design having three replicates of each group intended for analysis of variance type analyses (ANOVA) results in spatial pseudoreplication (Hulbert, 1984). Sampling one site within a stream five times before and after the implementation of a conservation practice with the intent of analyzing the data with ANOVA analyses is considered temporal pseudoreplication (Hulbert, 1984). Instead, three replicates for an ANOVA analyses would be achieved by sampling three control streams and three treatment streams. Additionally, sampling sites should be selected randomly from a pool of available sites when possible.

TABLE 2. Depiction of Replication Before and After Implementation of Conservation Practices Within Different Experimental Designs Used in Impact Assessments.

Experimental Design	Before	After
Replicated before–after-control-impact	□ □ □ ■ ■ ■	□ □ □ ■ ■ ■
Replicated before–after	■ ■ ■	■ ■ ■
Replicated space-for-time		□ □ □ ■ ■ ■
Unreplicated before–after-control-impact	□ ■	□ ■
Impact *vs*. control sites	□ □ □ ■	□ □ □ ■
Unreplicated before–after	■	■
Gradient		▪ ▪ ▪ ■ ■ ■

Note: Boxes represent control streams (unshaded boxes) and treatment streams (shaded boxes).

The use of replicated experimental designs with appropriate controls and treatments represents the ideal situation that can be logistically difficult to achieve. However, we feel investigators should strive to obtain the ideal situation first, and then accept alternative designs if logistical barriers are encountered. Replicated before–after-control-impact (BACI) designs are considered to be a powerful approach in evaluating the impacts of anthropogenic habitat alterations (Downes *et al.*, 2002). Replicated BACI designs involve sampling multiple control streams and multiple treatment streams simultaneously (or nearly so) before and after implementation of conservation practices (Table 2). Replicated before–after designs and replicated space-for-time designs are also acceptable experimental designs despite their implicit assumptions. Replicated before–after designs involve sampling multiple treatment streams before and after implementation of conservation practices (Table 2) and assume the observed change between the before and after sampling periods is due to the conservation practice, not temporal variation. Replicated space-for-time designs involve sampling control and treatment streams without any before data and assume that conditions within control streams represent conditions of treatment streams in the absence of the conservation practices (Table 2). The number of replicates to be sampled can be determined given an estimate of the variability of the response variables and the minimum change in response variables among experimental treatments that the investigators wish to detect (Green, 1979; Zar, 1984; Loftis *et al.*, 2001).

BACI designs were originally intended to be used in unreplicated studies having a single control and treatment stream (Green, 1979; Table 2). Unreplicated BACI designs are also referred to as paired watershed designs and evaluate the impact by sampling control and treatment streams multiple times before and after the implementation of the treatment (Table 2). An unreplicated before–after study would sample one treatment stream multiple times before and after implementation of a treatment (Table 2). A special case of the BACI design is the impact *vs*. control sites design (Stewart-Oaten and Bence, 2001) where the investigator samples one treatment stream and multiple control streams before and after implementation of the treatment (Table 2). The impact *vs.* control sites design is intended for situations where the investigator is unable to replicate the treatment, but can replicate the control. We consider unreplicated and unbalanced designs less desirable than replicated designs. However, unreplicated BACI designs and impact *vs.* control sites with an adequate temporal scale (i.e., a minimum of two years before and two years after data) would be preferable to unreplicated before–after studies because these experimental designs incorporate a control site (or sites) for comparison with the treatment stream.

Large scale studies similar to monitoring studies are also capable of providing useful information on the influence of conservation practices if the selection of sampling sites is conducted so a continuum of habitat conditions encountered are within the context of the *a priori* hypothesis. For example, one could examine the influence of streamside buffers by sampling sites with riparian buffer widths ranging from 0 (no buffer) to 100 m. This type of study would be considered a gradient design (Table 2), and its disadvantage is that causality can not be inferred from correlation. However, the gradient design is an effective tool that enables investigators who lack the opportunity or authority to control the implementation of conservation practices to document useful information about the potential effects of conservation practices.

All experimental designs discussed above are imperfect and have inherent flaws. It is feasible that valuable information could be obtained from impact assessments using alternative designs that we failed to consider. However, we feel success in evaluating ecological impacts does not depend on the experimental design alone, but selecting experimental designs on the basis of the hypothesis of interest and recognizing the limitations of the chosen experimental design in determining causality.

Assess the Habitat and Biological Characteristics With Quantitative and Repeatable Sampling Methods

Habitat characteristics include chemical, hydrological, geomorphological, and other physical descriptors of lotic ecosystems that form the space that living things occupy. We recommend that habitat and biological characteristics be measured at spatial and temporal scales that allow the investigator to explore the relationships between the habitat and the biota. For some habitat variables this might mean taking measurements on the same day the biological variables are measured (e.g., water depth), while other habitat factors that are not expected to fluctuate significantly during the year may only need to be measured once a year (e.g., sinuosity). Concurrent assessment of habitat and biological characteristics leads to more robust evaluations of conservation practices than evaluations that assess either type of characteristics alone (Maddock, 1999). Concurrent measurement of habitat and biota also enable the investigator to identify habitat factors that contribute to the biological changes and establish causal links to conservation practices (Simonson *et al.*, 1994). We are not recommending the measurement of all possible habitat variables, instead, we recommend that habitat variables are selected based on the hypothesized effects of the conservation practice on the habitat. We anticipate the selection of taxonomic groups (e.g., macroinvertebrate, fish, amphibian, or reptile) to evaluate will depend on the research interests of the investigator. We also recommend evaluating multiple taxonomic groups when possible, because different biota have different habitat criteria and will respond differently to habitat changes (Lake, 2001).

Use of quantitative and repeatable methods is necessary to assess how the habitat and biota change through time and space. We focus on the importance of this recommendation for habitat sampling because of the prevalence of the use of qualitative habitat measurements used by regulatory agencies in the U.S. Specifically, these qualitative protocols involve visual estimation of habitat variables such as water velocity, substrate types, and riparian habitat conditions. Qualitative assessments are quicker, but these methods should be avoided because the introduction of observer bias affects repeatability and objectivity (Somerville and Pruitt, 2004). Specifically, we recommend transect-based sampling (Gorman and Karr, 1978) using permanent transects for measurement of geomorphology, riparian habitat characteristics, and instream habitat. Additionally, systematically placing transects throughout a site reduces observer bias. Transect-based habitat measurements are widely and successfully used to characterize stream habitat conditions and ensure the repeatability and comparability of habitat data across space and time (Simonson *et al.*, 1994; Wang *et al.*, 1996; Shields *et al.*, 1998).

Use Multiple Sampling Techniques for Collecting Aquatic Organisms

All sampling techniques for aquatic organisms are biased towards the capture of certain sizes and types of organisms. The observed results of impact assessments may differ among sampling techniques as a result of sampling biases (Scarsbrook and Halliday, 2002). The use of more than one gear type is an effective way to account for sampling biases of individual techniques and obtain data of sufficient quality (Karr, 1999). We recommend the use of at least two sampling techniques for each taxonomic group. One does not need to devote the same level of effort with each gear type. The most effective sampling technique can be used as the primary sampling method, while the second or third could be used less intensively and considered supplementary. This recommendation is particularly important for community level assessments to ensure that stream communities are adequately characterized.

Standardize Sampling Efforts for Aquatic Organisms

The number and types of aquatic organisms captured is also influenced by the sampling effort. Typically, increasing the length of the sampling site and the number of collections made will result in an increase in species richness and abundance. Additionally, the length of a site is typically set equal to a certain number times the mean water surface width. This method of determining the site lengths may result in a variation in site lengths and sampling effort among streams if the mean water surface width differs among streams. We recommend standardizing the length of the sampling site and the number of samples to enable straightforward comparisons among experimental treatments and across time periods. Additionally, similar sampling sizes result in statistical analyses that are robust to the underlying assumptions (Downes *et al.*, 2002).

Previous work has shown that estimates of biological response variables vary with the length of sampling sites (Lyons, 1992; Li *et al.*, 2001). Results of these studies provide valuable information for monitoring studies that may sample one site within a large number of tributaries as part of a watershed-wide study. However, these results have less relevance for designing impact assessments because the focus of these studies is not the estimate of a response variable from one site, but the comparison between unimpacted and impacted sites. There is a tradeoff between site length and replication that investigators must consider when designing impact assessments as longer sites require more time to sample and may limit the number of sites (replicates) that can be obtained. Therefore, impact assessments would benefit more by devoting their sampling efforts among shorter length sites within more control and treatment streams to increase the power of the statistical analyses than they would be sampling longer length sites within fewer control and treatment streams. To ensure that site lengths are adequate, investigators could collect preliminary stream width measurements from the study sites first, and then multiply the mean stream width times a constant to establish a standard site length that will be appropriate for the majority of the sites given the time and resource allocations.

APPLYING THE GUIDING PRINCIPLES

We discuss below an example of how our recommendations were used to develop a sampling protocol for a five-year study assessing the influence of herbaceous riparian buffers (i.e., filter strips) on channelized headwater streams (i.e., drainage ditches). Additionally, sampling methodology is discussed in detail so the example also highlights equipment and techniques appropriate for use in wadeable streams in the eastern U.S. The study is one of the ecological assessments being conducted as part of the ARS CEAP Watershed Assessment Study. It began in May 2006 and funding for this study required that field work be conducted within the Upper Big Walnut Creek watershed in central Ohio.

Research Hypothesis

The following hypothesis was formulated to guide the experimental design: *Establishment of herbaceous riparian buffers adjacent to headwater drainage ditches of Upper Big Walnut Creek watershed will alter riparian habitat and geomorphology, which will in turn cause measurable changes in water chemistry, instream habitat, and the structure of stream communities (fishes, macroinvertebrates).* The hypothesis identifies the major classes of independent and dependent variables that must be evaluated.

Experimental Design and Selection of Sampling Sites

A replicated space-for-time experimental design was selected because the investigators lacked control of when the herbaceous riparian buffers were established and were not able to begin sampling before the buffers were installed. The investigators plan to use a two factor repeated measures ANOVA to compare differences in geomorphological, riparian, hydrological, chemical, and biological characteristics among: (1) drainage ditches without herbaceous riparian buffers (control), (2) drainage ditches with herbaceous riparian buffers (buffer treatment), and (3) streams with remnant forested riparian zones (minimally impacted stream). Treatments represent a range of environmental conditions from the worst case (i.e., the control treatment) to the best case (i.e., minimally impacted stream) management strategies for headwater streams in this region. Some physical and biological differences are expected to occur between channelized and unchannelized streams. However, this experimental design was developed with the intent of using the control and the minimally impacted stream treatments as opposing reference points for evaluating the effectiveness of the herbaceous buffers. Specifically, the evaluation criteria that will determine if herbaceous riparian buffers are effective is the prediction that habitat and biological conditions within the buffer treatment should become more similar to the minimally impacted streams than control streams, particularly with increasing time following the establishment of the buffers.

All sampling sites are located on private land and were selected after conducting site visits to evaluate them based on predetermined habitat criteria: (1) first or second-order headwater ditches or streams, (2) land use within the watershed is predominantly agriculture, (3) accessibility that allowed for establishment of two sites spaced at least 150 m apart. Additional habitat criteria used to create the treatment effect were: (1) replicates for the control were to be drainage ditches having narrow riparian zones (<15 m) with mostly herbaceous riparian vegetation, (2) replicates for the buffer treatment were drainage ditches having herbaceous riparian buffers on both streambanks that were established between 2003 and 2005 through the Conservation Reserve Enhancement Program, and (3) replicates for the minimally impacted streams were required to be unchannelized streams with mature woody vegetation >5 m tall on both streambanks.

A replicate in this study is a ditch or a stream. Drainage ditch treatments have three replicates each, but the minimally impacted stream treatment has only two replicates due to the difficulty in finding and obtaining permission to sample another unchannelized stream. Two 125 m long sites (subsamples) were established in each replicate and the sites are at least 150 m apart to ensure our sampling encompasses a representative range of habitat conditions within each replicate. Habitat and biological data from each site within a ditch/stream during each sampling period will either be composited or averaged to avoid pseudoreplication.

Sampling Methods

Six permanent transects spaced 25 m apart were established beginning at the downstream border of each site. Permanent transects were established with the use of wooden stakes driven into the ground at the points on both banks that represent estimated bankfull conditions (i.e., the point where the floodwaters would begin spilling over into the floodplain) (Figure 2). Riparian vegetation, geomorphology, and instream habitat are measured along each transect.

Riparian and geomorphological characteristics are measured once a year (Figure 2). Species composition and density of woody vegetation >1 m tall are measured in twelve 1 m $\times$ 10 m quadrats that begin at the water's edge and coincide with permanent transects (Figure 2). The presence and absence of herbaceous and woody vegetation within four height stratas (0-0.5 m, 0.5-2 m, 2-5 m, and >5 m) are noted in each quadrat. Riparian canopy cover is measured using a convex spherical densiometer while facing upstream in the middle of each quadrat and the wetted portion of the channel. Riparian zone widths of the drainage ditches were determined by calculating the straight line distance between coordinate measurements obtained at the water's edge and the edge of the agricultural fields (Figure 2) during geomorphology surveys. Woody vegetation cover and widths of the minimally impacted streams hindered the use of surveying methods for riparian width measurements in these sites. Therefore, riparian widths of the minimally impacted streams were obtained using aerial photos and geographic information systems (ArcGIS). Geomorphological variables including top bank width (i.e., the width of the channel at bankfull capacity), thalweg depth, channel cross-section area, gradient, and sinuosity are calculated from coordinate measurements collected from a minimum of nine points along each transect using either a Real Time Kinematic (RTK) system or an electronic total station (Figure 2). Both instruments provide the necessary level of accuracy and the total station is typically only used in the minimally impacted streams because the canopy cover within these streams prevents the use of the RTK system.

Measurements of instream habitat characteristics are obtained in the spring (April-May), summer (July-August), and fall (September-November) of each year (Figure 2). One measurement of wet width and four measurements of water velocity, depth, substrate, and instream habitat feature are obtained along each transect. Wet width is determined by measuring the distance between the left and right water's edge (Figure 2) with a tape measure. Water velocity is measured with an electromagnetic velocity meter. Water depths are measured with a stadia rod. The dominant substrate type and instream habitat features are visually identified at each point. This method enables the percent of substrate and instream habitat features to be calculated based on the number of times each substrate and instream habitat feature occurred in a site relative to the total number of points sampled. In contrast, qualitative estimates of

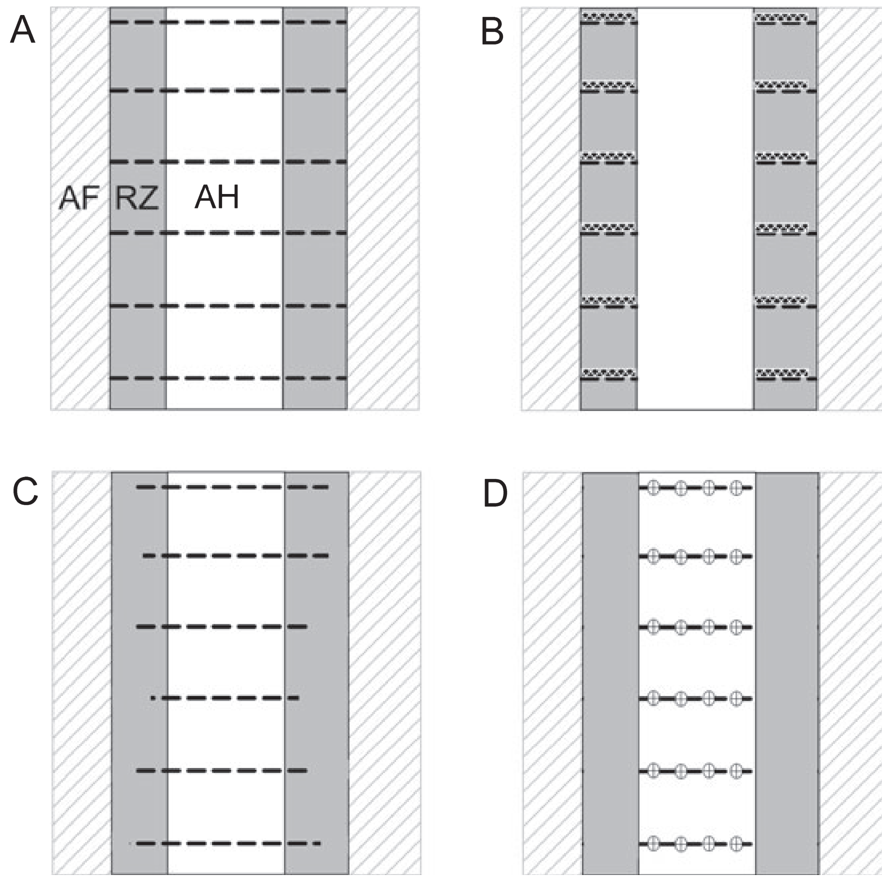

FIGURE 2. Schematic Depicting Locations of Six Permanent Transects Within a 125 m Long Site for Sampling Geomorphology, Riparian Habitat, and Instream Habitat (A). Transects encompass the Riparian Zone (RZ), Aquatic Habitat (AH), and end at the Agricultural Field (AF). Riparian habitat characteristics are measured along each transect and within 1 m × 10 m quadrats located upstream of the transects (B). Geomorphology measurements are taken within the left and right top banks along the transects (C). Measurements of instream habitat characteristics are collected at four equidistantly spaced points along each transect within the aquatic habitat (D).

percent substrate types involve an investigator walking a reach and then estimating the percent of substrate and instream habitat features. Instantaneous discharge is measured using a wading rod and current meter at an additional transect within each site.

Water chemistry characteristics are measured *in situ* or by analysis of grab samples. Water samples collected from weekly grab samples obtained from May to December are analyzed for nutrients, herbicides, and total suspended solids. *In situ* measurements of dissolved oxygen, temperature, pH, and conductivity are obtained with a multiparameter meter from one location within a site three times a year concurrently with instream habitat, fish, and macroinvertebrate sampling. Grab samples for turbidity measurements are collected in conjunction with *in situ* measurements.

Fishes and macroinvertebrates are sampled three times a year concurrently with instream habitat and *in situ* water chemistry measurements. Block nets are set at the upstream and downstream borders of the sites prior to sampling fishes and macroinvertebrates. Fishes are sampled with a backpack electrofisher and seine. Electrofishing begins at the downstream border and proceeds upstream. After electrofishing, five seine hauls are conducted and distributed throughout a site. Pools and slow flowing areas are sampled with a haul and fast flowing riffle areas are sampled using the seine as a block net and kicking into the seine. The seining effort is standardized by ensuring each haul or kick sample encompasses a 4 m length of shoreline. This provides flexibility to use the best methods for sampling a given habitat type and ensures that sampling efforts are standardized.

Macroinvertebrates are sampled with dipnet and surber samples after completion of fish sampling. Sampling is conducted to ensure that sampling efforts are distributed throughout the site and that collections are obtained from each habitat unit (i.e., pools, riffles, runs) present in a site. Three dipnet samples are collected from areas that are either too deep or have too slow of a water flow for effective surber sampling. A dipnet sample consists of a 1 m long dipnet sweep that samples macroinvertebrates in the benthos and water column. Three surber samples are collected from riffles and shallow runs.

Preliminary Results

Differences in selected response variables from the first year of the study (2006) were examined with single factor ANOVA (riparian habitat and geomorphology) or two factor repeated measures ANOVA (instream habitat, water chemistry, and fish community response variables). Preliminary results confirm the site selection criteria as the minimally impacted streams had greater ($p < 0.05$) riparian widths, canopy cover, woody vegetation density, and sinuosity than the ditches with and without herbaceous riparian buffers (Table 3). Additionally, drainage ditches with and without herbaceous riparian buffers had greater ($p < 0.05$) thalweg depths and top bank widths than the minimally impacted streams (Table 3). Water velocity was greater ($p < 0.05$) in the minimally impacted streams than ditches with and without herbaceous riparian buffers (Table 3). No differences ($p > 0.05$) in other instream habitat, water chemistry, or fish community response variables were observed (Table 3). However, general trends in water chemistry and fish community variables among treatments were as expected. Water temperature, turbidity, and percent omnivores (percent of fishes that eat plants and animals) were greater in the drainage ditches with and without herbaceous riparian buffers than the minimally impacted streams (Table 3). Fish species richness and Percidae abundance (number of darters) were the greater in the minimally impacted streams than drainage ditches (Table 3). The high variability in water chemistry and fish community response variables likely hindered detection of significant differences among means in this preliminary analysis. Future analyses using five years of data will have greater sample sizes that will improve the ability to detect differences among treatments. These preliminary results represent the starting point in this evaluation of herbaceous riparian buffers and future analyses will examine how the habitat and biota within the treatments change over time.

CONCLUSIONS

Guidelines for designing impact assessments to assess the ecological responses of streams to conservation practices have been developed based on our experiences and others. The guidelines were intended to provide guidance for others investigating the influence of restoration and other anthropogenic habitat alterations with impact assessments in small to

TABLE 3. Mean (SD) Riparian Habitat, Geomorphology, Instream Habitat, and Fish Community Response Variables Among Drainage Ditches Without Herbaceous Buffers (control), Drainage Ditches with Herbaceous Buffers (buffer treatment), and Unchannelized Streams With Forested Riparian Buffers (minimally impacted) Within the Upper Big Walnut Creek, Ohio, 2006.

	Control	Buffer Treatment	Minimally Impacted
Riparian habitat			
Riparian width (m)	8.2 (2.4) C	49.0 (28.5) B	99.4 (14.9) A
Canopy cover (%)	6.7 (6.5) B	6.9 (10.1) B	71.6 (19.5) A
Woody vegetation density (#/m^2)	0.2 (0.2) B	0.2 (0.1) B	0.9 (0.1) A
Geomorphology			
Thalweg depth (m)	1.8 (0.2) A	1.6 (0.4) A	0.8 (0.1) B
Top bank width (m)	9.0 (1.2) A	8.9 (1.1) A	4.8 (0.0) B
Sinousity	1.0 (0.0) B	1.0 (0.1) B	1.6 (0.2) A
Instream habitat			
Water depth (m)	0.12 (0.04) A	0.13 (0.09) A	0.13 (0.02) A
Water velocity (m/s)	0.03 (0.05) B	0.01 (0.02) B	0.09 (0.02) A
Wet width (m)	2.00 (0.48) A	1.74 (1.00) A	2.40 (0.30) A
Water chemistry			
Water temperature (°C)	18.8 (7.2) A	17.2 (6.4) A	14.5 (6.3) A
Dissolved oxygen (mg/l)	8.4 (3.2) A	8.8 (3.8) A	9.8 (1.9) A
Turbidity (NTU)	69.7 (50.4) A	52.2 (47.3) A	31.0 (28.7) A
Fish community			
Species richness	7.9 (3.8) A	7.0 (5.4) A	8.7 (1.2) A
Percent omnivores	28.4 (27.6) A	32.4 (33.4) A	4.9 (4.3) A
Percidae abundance	27.4 (26.7) A	24.2 (21.5) A	95.5 (44.2) A

Note: Different letters within a row indicate a significant ($p < 0.05$) difference among treatments.

medium-sized lotic ecosystems. These guidelines have been used to guide the design of impact assessments of conservation practices in ARS CEAP watersheds in Ohio and Indiana. Specifically, these ongoing studies are evaluating the influence of herbaceous riparian buffers, nutrient management, pesticide management, and relative influence of physical and chemical habitat characteristics on fish and macroinvertebrate communities in agricultural drainage ditches (King *et al.*, 2008; Smiley *et al.*, 2008). Similar guidelines were used in the design of CEAP-related research in Mississippi (Shields *et al.*, 1998, 2005, 2006b). Additionally, our paper provided information that should facilitate cross-disciplinary assessments of conservation practices because it highlights for soil scientists, engineers, hydrologists, geomorphologists, and individuals from other disciplines the framework used by many ecologists for evaluating the impacts of anthropogenic habitat alterations on streams.

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